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Regeneration of *Acacia* and *Kennedia* from soil stored seed following an autumn fire in jarrah (*Eucalyptus marginata*) forest

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Abstract

Regeneration of species of *Acacia* and *Kennedia* following a fire in autumn was studied in the understorey of jarrah forest near Dwellingup. A total of 4 species of *Acacia* and 2 species of *Kennedia* were recorded in 46 plots assessed prior to burning. Six months after burning all except 1 species (*Acacia pulchella*) were recorded in more plots than before the fire. *Kennedia coccinea* exhibited the greatest increase in distribution following burning. Total seedling numbers did not differ significantly between plots burnt at low intensity (<300 kWm^{-1}) and plots burnt at greater intensity, including full scorch of the overstorey. Between August 1980 and February 1983 similar changes in plant density were recorded in fenced exclosures and adjacent unfenced areas at 5 sites indicating that grazing had no major impact on populations over this period. A broadscale survey of the 2 200 ha study area 1 year after the fire indicated that about 10% of the area carried dense thickets of *A. pulchella* and *A. celastrifolia* while a further 15% carried thickets of *Acacia drummondii*.

Introduction

Understoreys of native legumes are a distinctive feature of jarrah (*Eucalyptus marginata*) forest throughout the range of this forest type. Most of the common legumes are obligate seed regenerators which re-establish from seed stored in the top few centimetres of the soil profile (Shea *et al* 1979, Monk *et al* 1981). Hardseededness (Gill 1981) is common amongst the legumes with the result that there is little or no regeneration in the absence of fire or mechanical disturbance of the soil.

Germination of soil stored seed following fire is influenced by a number of factors (Gill 1981) including the abundance and viability of seed, the depth of seed burial, the response of different species to heat and the extent of soil heating (Shea *et al* 1979, Monk *et al* 1981, Floyd 1976). The extent of soil heating depends on the physical characteristics and moisture content of the soil (Aston & Gill 1976) and the heating characteristics of the fire, in particular the quantity of fuel consumed (Knight 1981). Fires during summer and early autumn provide ideal conditions for soil heating because soils are dry and litter fuel is entirely consumed (Christensen *et al* 1981).

In jarrah forest, regeneration of dense legume thickets has often been observed following summer fires of moderate to high intensity (Peet 1971, Peet & Van Didden 1973, Shea *et al* 1979, Monk *et al* 1981) but has rarely been associated with the spring fires of low intensity which are periodically used to reduce accumulations of fuel. Manipulation of the season and intensity of prescribed burning has been proposed as a means of promoting the regeneration of dense thickets of legumes for purposes including the maintenance of animal habitat (Christensen 1980), and the establishment of understoreys resistant to *Phythophthora cinnamomi*, the causal agent of jarrah dieback (Shea *et al* 1979). The factors which determine the regeneration and subsequent growth of native legume species need to be understood if such treatments are to be implemented on an operational scale.

This paper examines the post-fire regeneration of species from 2 genera of native legumes, *Acacia* and *Kennedia*, which are commonly found on a wide range of sites in jarrah forest. Aspects examined are: regeneration in relation to species occurrence prior to burning; comparison of the density of regeneration in forest subject to 2 levels of crown scorch (an index of the intensity of fire); and the effect of grazing exclosures on changes in plant numbers during the 2.5 years following seedling germination.

Study area

The study was undertaken in the 2 200 ha Hakea forest block, 25 km SE of Dwellingup (34° 43'S, 116° 04'E). Six of the vegetation complexes described and mapped for the Darling System by Heddle *et al* (1980) occur within the area, as follows (Fig. 1); Dwellingup complex, medium-high rainfall (1074 ha); Yarragil complex, maximum swamp development (450 ha); Yarragil complex, minimum swamp development (139 ha); Murray-Bindoon complex (211 ha); Pindalup-Yarragil complex (172 ha); and Cooke complex (154 ha).

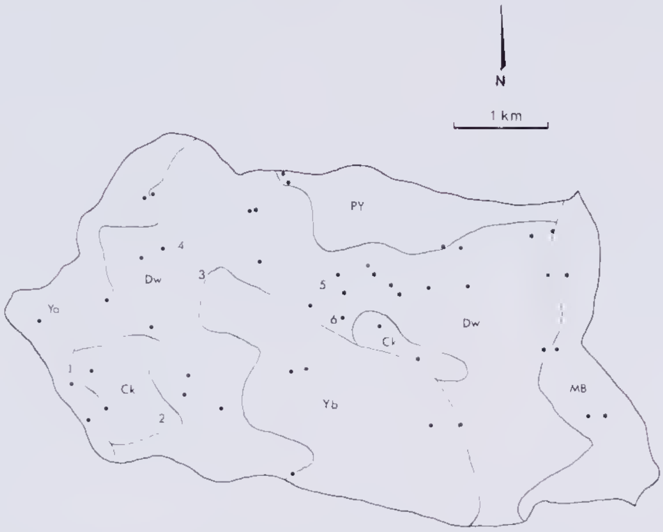

Figure 1 Map of Hakea Block showing location of plots (.) and grazing exclosures (1 — 6). Vegetation complexes (Heddle *et al* 1980) represented in the area are Dwellingup (Dw), Yarragil with minimum swamp development (Ya), Yarragil with maximum swamp development (Yb), Murray-Bindoon (MB), Pindalup-Yarragil (PY) and Cooke (Ck).

Vegetation complexes correspond to the major landform units of Churchward & McArthur (1980), and typically combine several of the more specific site vegetation types defined by Havel (1975).

The dominant vegetation on the lateritic uplands that comprise about half the study area (Dwellingup and Cooke complexes) is open forest of jarrah and marri (*Eucalyptus calophylla*), 20-30 m in height, with an understorey of low shrubs. The forests of the other vegetation complexes may also contain jarrah and marri but are generally less than 20 m in height and are distinguished by the presence of other eucalypt species including yarri (*Eucalyptus patens*) and wandoo (*Eucalyptus wandoo*). The area of Yarragil complex in the southern part of the block is dominated by an extensive *Banksia littoralis* swamp.

Fire history records available from 1938 onwards indicate that only small parts of Hakea block have been burnt by wildfires, but that the entire area was subject to fuel reduction burning in spring 1968 and again in spring 1974.

Methods

Pre-fire assessment

The number of plants of *Acacia* and *Kennedia* species was counted in each of 46 permanently marked 20 × 20 m plots during January 1980. These plots were randomly located within the study area (Fig. 1). Contacts with each species were also recorded from 50 point samples following the technique of Levy & Madden (1933). Two legume species common in the area (*Acacia preissiana* and *Bossiaea ornata*) were not included in the study because they commonly regenerate from rootstock after fire and it was considered that this might create difficulty in determining the true extent of regeneration from seed.

Litter fuels samples were collected from four 1 m^2 quadrats in each plot for determination of oven dry weight. Understorey fuels were not sampled at each plot, but would probably have contributed an additional 1-2 t ha^{-1} to the total quantity of fuel on upland sites (Sneeuwjagt & Peet 1985).

Samples of litter fuel and the 0-5 cm horizon of the soil were collected at a number of upland sites on the morning immediately prior to burning of the study area. Samples were subsequently oven dried for moisture content determination.

The fire

The study area was burnt on 22 March 1980. The first widespread rain following the summer drought had fallen 2 days previously with 3 mm being recorded in a portable rain gauge located at the eastern end of the study area. On the day of the fire, weather conditions were recorded at the study area and at a nearby fire lookout tower (Table 1).

The area was ignited by an aircraft dropping incendiaries on a grid pattern; ignition commenced at 1500 hrs and continued for 2 hours, after which time fires continued to burn into the evening until about 2100 hrs. Around the perimeter of the area the intensity of fires was generally low ($<$300 kWm^{-1}) but considerably higher intensities were observed from the aircraft within the centre of the area.

Table 1

Summary of fuel, soil moisture and weather conditions at the Hakea study area on the day of burning (22 March 1980).

Weather	
Maximum temperature (°C)	27
Minimum relative humidity (%)	30
In-forest wind speed at 1600 hr (kmh^{-1})	3
Tower wind speed at 1600 hr (kmh^{-1})	11
Tower wind direction	ESE
Fuel	
Litter fuel quantity (t/ha) mean	10.5
range	3.2-24.9
SE	0.8
Minimum litter fuel moisture content range (%)	9-10
Soil	
Soil Dryness Index[A]	c 1600
Moisture content of 0-5 cm horizon (%) mean	4
range	3-7

[A] Refer to Mount (1972) and Burrows (1987).

Post-fire ossessment

Several months after the fire the height of crown scorch was measured on a sample (10-20) of dominant trees in each plot to provide an estimate of the intensity of fire. Based on the relationships reported by Burrows (1984), plots with scorch height ≤15 m were estimated to have burnt with an intensity <300 kWm^{-1}, which is regarded as low (Cheney 1981).

Acocia and *Kennedia* regeneration within the 20 × 20 m plots was initially assessed in August 1980. The number of seedlings of each species was counted on four 10 m^2 circular quadrats which were located in the corners of each plot. This sampling procedure was adopted because the very high numbers of seedlings in some plots made it impractical to count seedling numbers over the entire plot area. Plant densities before and after burning were expressed in terms of plant/m^2. Point sampling was repeated in each plot 2.5 years after the fire.

During early August 1980 grazing exclosures were constructed at 6 sites (Fig. 1) where germination of *A. drummondii* and *A. pulchello* was widespread. At each site an area of about 0.5 ha was fenced with wire mesh to 1.5 m height, and seedling numbers counted on 40 permanently marked 1 m^2 quadrats within the exclosure and on 40 adjacent quadrats outside the exclosure. Quadrats were recounted in February 1983. Mean seedling numbers in 1981, and changes in seedling numbers between 1981 and 1983 were compared between fenced and unfenced quadrats at each site using a *t* test.

The broadscale distribution of *Acocio* and *Kennedio* species was surveyed 12 months after the fire. Survey points were located every 100 m along a series of transects which ran parallel at 400 m intervals across the study area. Plant numbers were counted on a 10 m^2 circular quadrant at each point. The distribution of each species was then plotted at 1:50 000 scale and overlayed onto a map of the vegetation complexes of Heddle *et ol* (1980) for tallying of species occurrence by complex. Expected frequencies were calculated according to the hypothesis that species were present in equal proportion in each complex. Observed and expected frequencies were compared using the likelihood ratio (Zar 1974). Several species recorded only at a few points were not subject to this analysis.

Results

Four species of *Acocia* and 2 species of *Kennedio* were recorded in plots prior to burning (Table 2). The most common species, *Acocio drummondii*, occurred in 17 plots while the remainder were found in 1-9 plots.

Six months after the fire all species, except *Acocio pulchello*, occurred in more plots than before the fire; *A. pulchello* was recorded in 9 plots before the fire but only 7 afterwards (Table 2). The Acacias typically regenerated within plots where they had been recorded prior to burning and in 1-4 additional plots. Two species which had not been recorded before the fire, *Acacio alato* and *Acacia extenso*, each regenerated in 1 plot following the fire.

Kennedio coccinea was considerably more widespread following the fire being recorded in 26 additional plots, while *Kennedio prostrata* regenerated in 5 additional plots (Table 2).

In a few instances species did not apparently regenerate in all plots where they had been present prior to burning. *A. drummondii. Acocia celastrifolio* and *K. coccinea* each disappeared from 1 plot after the fire, while *A. pulchello* disappeared from 3 plots.

The relationship between the number of plants in plots before and after burning varied between species. The density of regeneration of the 2 most common *Acacios* was strongly correlated to the preburn densities of these species within plots (*A. drummondii* $r = 0.756$, $N = 21$, $P < 0.001$; *A. pulchello* $r = 0.718$, $N = 9$, $P < 0.01$). A similar trend was apparent for several other species of *Acacio* although this could not be verified statistically because they occurred in too few plots. On the other hand both species of *Kennedia* mostly regenerated in plots where they had not been recorded prior to burning and no such relationships were apparent.

Numbers of seedlings in plots burnt at low intensity (<300 kWm^{-1}) were not significantly different from those observed in plots burnt at higher intensities, including a number subject to complete scorch of the overstorey canopy (Table 3).

Table 2

Occurrence of legume species within plots before and 6 months after burning.

	No. of plots in which each species was:			
Species	Not recorded before or after burning	Recorded before but not after burning	Not recorded before but recorded after burning	Recorded both before and after burning
A. alata R. Br.	45	0	1	0
A. celastrifolia Benth.	39	1	2	4
A. drummondii Lindley	25	1	4	16
A. extensa Lindley	45	0	1	0
A pulchella R. Br.	36	3	1	6
A. urophylla Benth. ex Lindley	41	0	2	3
K. coccinea Vent.	15	1	26	4
K. prostrata R Br	40	0	5	1

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Table 3

Comparison of the density of seedling regeneration of all legume species in plots with levels of crown scorch ≤15 m (estimated fire intensity <300 kWm^{-1}) and >15 m or fully scorched (estimated fire intensity >300 kWm^{-1}).

Variable		Scorch height ≤15 m	Scorch height >15 m
Total no. of plots in class		15	31
No. of plots containing legume seedlings		15	30
No. of seedlings m^{-2}	mean	1.32[1]	0.958[1]
	SD	1.110	1.653
	range	0.025-3.725	0-6.500

[1]Not significantly different at 0.05 level.

Table 4

Mean frequency (%) of legume species in 46 plots before and 2.5 years after burning determined by Levy point sampling.

Species	Mean frequency (%)	
	Before burning	2.5 years after burning
A alata	0	0
A celastrifolia	0.043	0.826
A drummondii	0.348	3.391
A extensa	0	0
A pulchella	2.000	2.860
A urophylla	0	0.261
K. coccinea	0.043	0.130
K prostrata	0	0.043

No significant relationship was apparent between the number of seedlings regenerated and the quantity of litter fuel in the plot (Fig. 2).

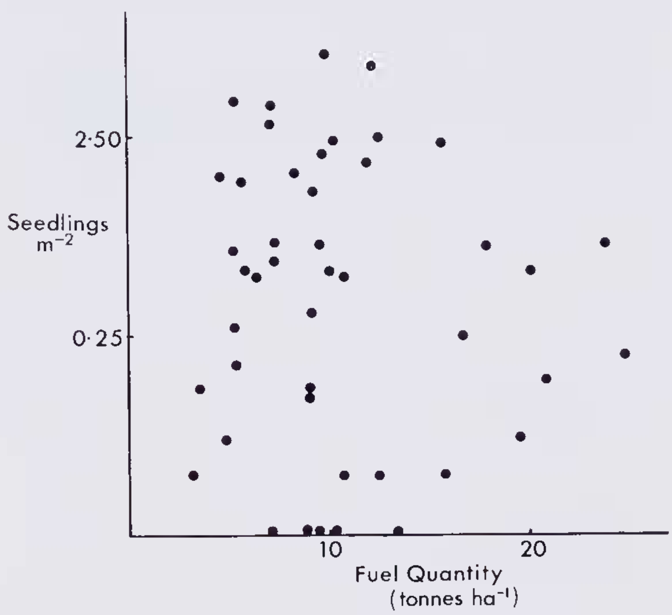

Figure 2 Number of seedlings per m² in August 1980 in relation to the quantity of litter fuel in each plot at the time of burning.

Pre- and postfire frequencies were calculated for each species from the Levy point sampling (Table 4). Within 2.5 years the frequency of contacts with each species had returned to prefire levels, and substantial increases were demonstrated for *A. drummondii*, *A. celastrifolia*, *Acacia urophylla* and *K. coccinea*.

At each of the 6 sites where exclosures had been fenced the density of *Acacias* in fenced and unfenced areas did not differ significantly at the time of the initial assessment in August 1981 (Table 5). Between August 1981 and February 1983 the density of plants declined to a similar extent in both fenced and unfenced quadrats at 4 of the sites. At the other 2 sites plant numbers increased over the same period: at site 1 plant numbers in the unfenced quadrats increased; at site 3 plant numbers in the fenced quadrats increased. Only at site 3 did the change in plant numbers differ significantly between fenced and unfenced quadrats. Changes in plant density were of a similar magnitude at each site irrespective of the location within the study area.

All of the species recorded in the plots were also encountered during the broadscale survey conducted 12 months after the fire at generally similar relative levels of frequency (Table 6). A further species (*Acacia divergens*) was also encountered locally in swampy areas where no plots had been established. The hypothesis of uniform relative frequency in all vegetation complexes was sustained for *K. prostrata* and *K. coccinea* but not for any of the common *Acacias* (Table 6).

Data from the broadscale survey were used to estimate the proportion of the study area which had sufficient density of the larger legumes (*A. pulchella* and *A. celastrifolia*) to facilitate development of dense thickets. For the purposes of this estimate it was assumed that a density ≥ 1 plant/m^2 of either species 1 year after the fire indicated potential for thicket development. On this basis plant densities were estimated to be sufficient for thicket development over about 10% of the study area. Using the same plant density criteria a further 15% of the area carried thickets of *A. drummondii*. Several of the other species grow to sufficient size but do not typically regenerate as thickets in jarrah forest (*A. urophylla*) or are restricted to localized areas, usually in swamps (*A. divergens*, *A. alata*, *A. extensa*).

Table 5

Mean numbers of *A. drummondii* and *A. pulchella* recorded in fenced and unfenced areas at 6 sites (40 1m^2 permanent quadrats per fencing treatment at each site).
See Fig. 1 for site locations.

Site	Fencing treatment	No. of plants m^{-2} in August 1980	Significance[1]	Change in number m^{-2} between August 1980 and February 1983	Significance[2]
1	fenced	16.7	t = 0.095	-0.10	t = 1.166
	unfenced	16.3	NS	+1.08	NS
2	fenced	20.9	t = 0.421	-0.75	t = 0.540
	unfenced	17.8	NS	-1.10	NS
3	fenced	24.3	t = 0.691	+1.48	t = 2.116
	unfenced	29.0	NS	-0.56	p < 0.05
4	fenced	20.8	t = 0.585	-0.59	t = 1.39
	unfenced	24.3	NS	-0.97	NS
5	fenced	11.5	t = 0.850	1.60	t =0.860
	unfenced	14.2	NS	1.00	NS
6	fenced	13.1	t = 0.700	-1.75	t = -0.093
	unfenced	15.6	NS	1.68	NS

[1] Comparison of plant density in fenced and unfenced quadrats at August 1980.

[2] Comparison of change in plant density in fenced and unfenced quadrats from August 1980 to February 1983.

Table 6

Frequency of legume species within the 6 vegetation complexes represented in the Hakes study area, based on data from 537 sample points. Figures in parentheses indicate the frequency expected if representation was proportional to the area of each complex.

Vegetation complex	Area within study (ha)	Observed and (expected) frequency within each complex for:								
		A.alata	*A.celastrifolia*	*A.divergens*	*A.drummondii*	*A.extensa*	*A.pulchella*	*A.urophylla*	*K.coccinea*	*K.prostrata*
Dwellingup med-high rainfall	1074	0	6(13.1)	0	60(73.9)	0	28(30.6)	10(10.2)	49(49.0)	14(16.0)
Yarragil (maximum swamp development)	450	0	12(5.5)	1	44(31.1)	0	7(12.9)	4(4.3)	20(20.7)	5(6.8)
Yarragil (minimum swamp development)	139	0	1(1.7)	0	12(9.6)	5	9(4.0)	5(1.3)	7(6.4)	2(2.1)
Murrary-Bindoon	211	1	1(2.6)	0	24(14.7)	0	2(6.1)	2(2.0)	12(9.7)	5(3.2)
Pindalup-Yarragil	172	1	0(2.1)	0	2(11.8)	0	6(4.9)	0(1.6)	7(7.9)	5(2.6)
Cooke	154	0	7(1.9)	0	10(10.8)	0	11(4.5)	0(1.4)	6(7.1)	2(2.3)
Total	2 200	2	27	1	152	5	63	21	101	33
G Statistic	-	ND	24.54	ND	25.29	ND	18.69	12.49	1.27	3.43
Significance level	–	ND	<0.001	ND	<0.001	ND	<0.005	<0.05	>0.05	>0.05

ND not determined

Discussion

Six months after burning all of the legume species examined in this study, except *A. pulchella*, were recorded in a greater number of plots than before the fire. There were relatively few examples where species were recorded in plots before, but not after the fire, and these can probably be attributed to the sampling technique adopted for the post-fire assessment. The density of *Acacia* populations has been observed to decline rapidly in the initial 4 years after fire (Monk *et al* 1981). Most, if not all species would have been less widespread at the time of the pre-fire assessment in 1980 than in the period soon after the 1974 fire.

The most pronounced increase in apparent distribution following burning was displayed by *K. coccinea* and it is likely that this species, being shortlived, had declined to a greater extent in the interfire period than had the longer lived, woody *Acacias*. Species richness and diversity in jarrah forest generally decline when the period between fires exceeds about 6 years due to the senescence of fireweeds and smaller herbaceous species (Bell & Koch 1980).

The intensity of fire, as indicated by crown scorch height, did not appear to influence the extent of germination. Peet (1971) found that the contribution of leguminous species to the understorey, based on projected foliage cover, only varied slightly between areas burnt at a wide range of intensity during a summer wildfire; legumes contributed 38% in defoliated forest, 40% in fully scorched forest and 33% in forest subject to lesser crown damage. Legume contribution was, however, only 20% in similar forest which had been burnt in spring at low intensity. This is consistent with the hypothesis that germination of soil stored seed is governed primarily by fuel and soil moisture at the time of burning, rather than the acutal behaviour of the fire (Christensen & Kimber 1975). However where the intensity of fire is sufficient to seriously damage the forest canopy it is likely that understorey thicket development may be enhanced by the temporary reduction is competition. Peet (1971) reported that the total understorey cover (all species including legumes) did in fact increase proportionally with the level of damage to the canopy. Cochrane (1968) also reported that canopy characteristics had a primary regulating role on the regeneration of understorey species following fire in dry sclerophyll forest in Victoria.

No significant influence of fuel quantity on legume germination was established in this study, despite the probable importance of this factor (Christensen and Kimber 1975). Observations following the fire indicated that a large proportion of litter fuel had been consumed, so that differences between the quantity measured prior to burning and the quantity actually consumed by the fire were not great. The most likely explanation for the lack of any clear relationship is that it was obscured by variation between plots in the quantity of seed, depth of burial and heat response characteristics of different species. A further factor potentially influencing germination response was rainfall before the fire. Although the amount of rain (3 mm) was small, it may have been sufficient to dampen the surface layer of soil and restrict penetration of heat to deeper levels (2-4 cm) where much of the seed may be located (Shea *et al* 1979).

The similar changes in numbers of plants in fenced and unfenced areas at 5 sites suggests that grazing did not have a major impact on legume population densities after August 1980. However, the impact of grazing on population densities prior to August, and possible effects on the height or form of individual plants were not determined. Increases in number of plants at 2 sites after August 1980 would have been as a result of later germination. These results contrast with results reported by Shea *et al* (1979) and Christensen (pers. comm.) where heavy grazing of legumes has been observed particularly on small burnt areas and close to unburnt edges. The large size of the area burnt at Hakea and the fact that much of the surrounding forest had been burnt in the previous 12 months may have acted to disperse the potential grazing pressure.

The 2 species of *Kennedia* were ubiquitous throughout the study area but the *Acacias* were not present with equal frequency in all of the mapped vegetation complexes, and several were restricted to a few locations. In some cases this reflects a consistent link between species occurrence and site characteristics. Several of the legumes encountered at Hakea were used as specific indicator species in the site classification scheme developed by Havel (1975); these include *A. alata* (Type C), *A. extensa* (Type W), *A. urophylla* (Types S, T, Q). *A. drummondii* is characteristic of Type O but was not selected as an indicator species. Each of the vegetation complexes mapped by Heddle *et al* (1980) comprise several of Havel's site vegetation types and so the distribution of individual legume species would be linked to the occurrence of specific types within the complex. The limited presence of *A. urophylla* in the Yarragil (max. swamp development) complex would not be expected on the basis of known site vegetation occurrence of this species, but may simply reflect the limitations in the method used to combine the maps of species occurrence with those of the vegetation complexes. The 2 most common *Acacias* (*drummondii*, *pulchella*), were recorded in all vegetation complexes but with unequal relative frequencies in each complex; further investigation of species/site relationships would be required to adequately explain this pattern of distribution.

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